CHAPTER 7

METHODS OF BENEFIT MEASUREMENT

In the two preceding chapters, we have spoken of benefits in a rather general sense not specifying where they come from or how they might be measured in practice. In Chapter 5, for example, we assume the existence of a benefit function for ecosystem recovery and examine how a decision on pollution control is affected by the dynamics of recovery and the uncertainties surrounding it. In this chapter we look behind the benefit function. What kinds of benefits are provided by aquatic ecosystems, and how might they be measured? Here we take up the discussion begun in Chapter 1 drawing upon the classification of benefits and measurement approaches suggested there.

I. Aquatic Ecosystems as an Input to Production

Aquatic ecosystems function as an input to production whenever changes in an ecosystem's characteristics affect the costs of providing a good or service. For example, the number of wetland acres available as a habitat for fish may influence the cost of harvesting commercially valuable species. The quality of water withdrawn from rivers and lakes for municipal water supplies and irrigation determines the cost of subsequent water treatment and level of agricultural productivity. Finally, just as air pollution may lead to the chemical deterioration of materials, diminished water quality can lead to the corrosion of household appliances and industrial equipment. Valuing the benefits from improved environmental quality when the environment acts as an input to production is the focus of this section. We critically review a number of earlier studies in the area and go on to suggest (and illustrate) some improvements.

We focus on the examples identified in Chapter 1: supply of clean water and harvest of commercial species. Consider the former. Wetlands reduce the cost of water treatment by removing or settling pollutants. This can be represented as a shift in a marginal cost or supply curve along a given demand curve. An environmental improvement, such as provision of additional wetlands, would then involve a supply shift down and to the right, as from S to S' in Figure 1, where the shaded area between old (S) and new (S') supply curves indicates the net welfare gain, the change in consumer and producer surplus.

This is probably a typical case, but others are possible--and, it turns out, relevant to some of the existing literature. One, in particular, is worth noting. Suppose the new cost or supply curve is simply the horizontal axis. In other words, creation of the wetlands completely eliminates the need for human inputs, at least up to a point (represented by Q" on Figure 2). Then the welfare gain, illustrated in the figure, is the shaded area between old and new supply curves up to the point (Q' on the figure) where demand equals the old supply and between demand and new supply thereafter (up to Q"). Note that this is less than the area between the two supply curves. Beyond Q', consumer willingness-to-pay for water is less than the old cost of treatment so that the latter is no longer relevant.

This same point is made more dramatically in Figure 3. There the old cost of treatment or supply curve lies everywhere above the demand curve. The benefit of the environmental improvement, represented as a shift in the supply curve to coincide with the horizontal axis, is then simply the area under the demand curve (up to Q"). The area between the two supply curves, which is just the area under the old curve, or the cost of providing treatment in the absence of the wetlands, would overstate the benefit of having the wetlands for this purpose.

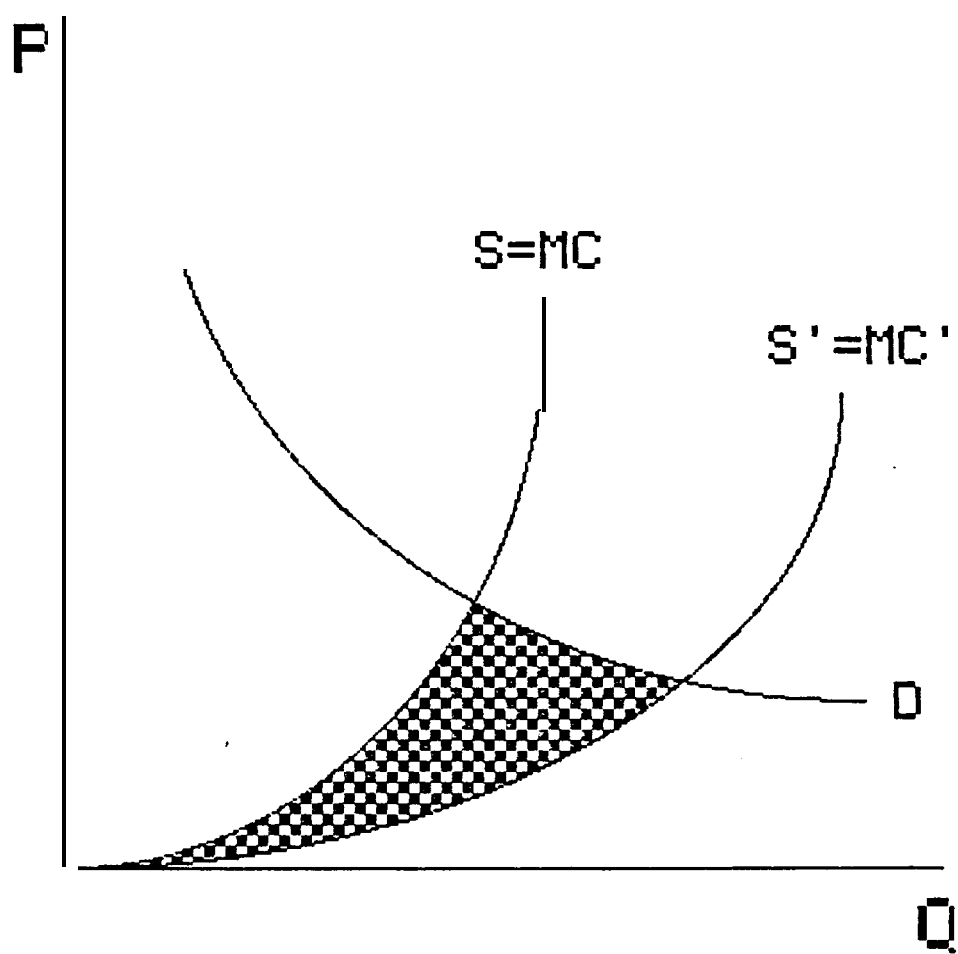# FIGURE 1

WASTE ASSIMILATION
BENEFIT PROVIDED BY
THE ECOSYSTEM

FIGURE 2

WASTE ASSIMILATION
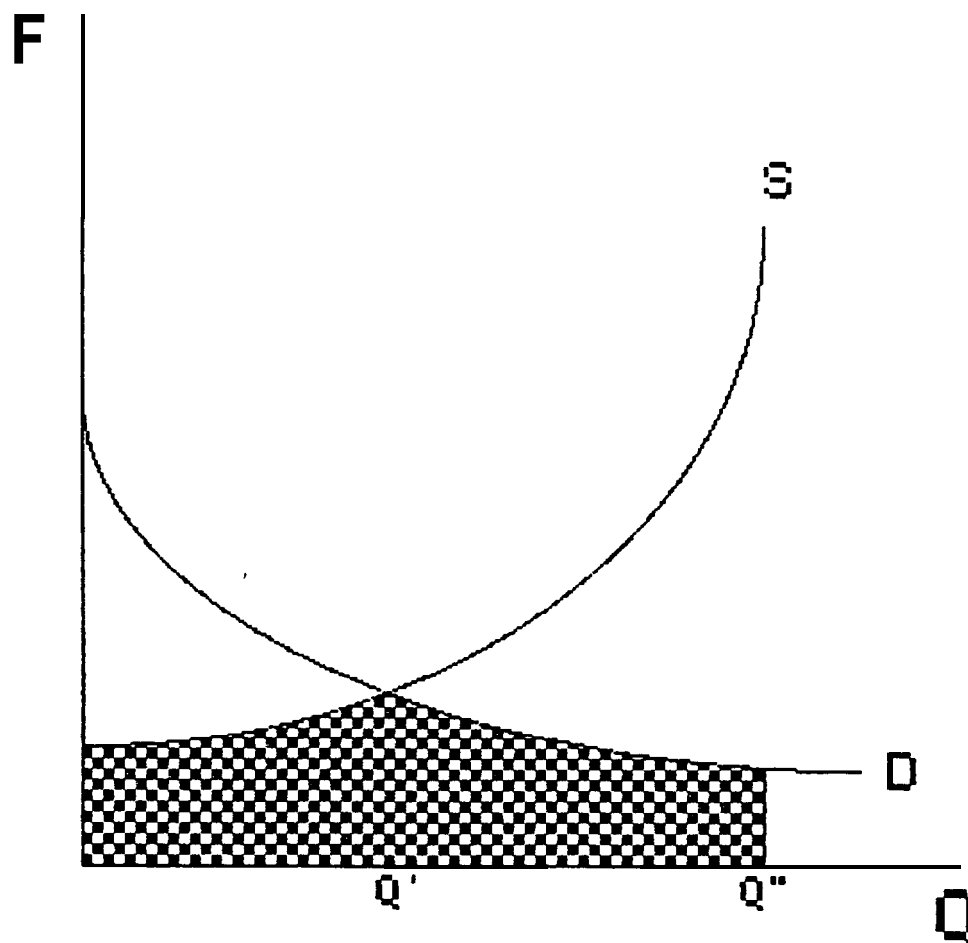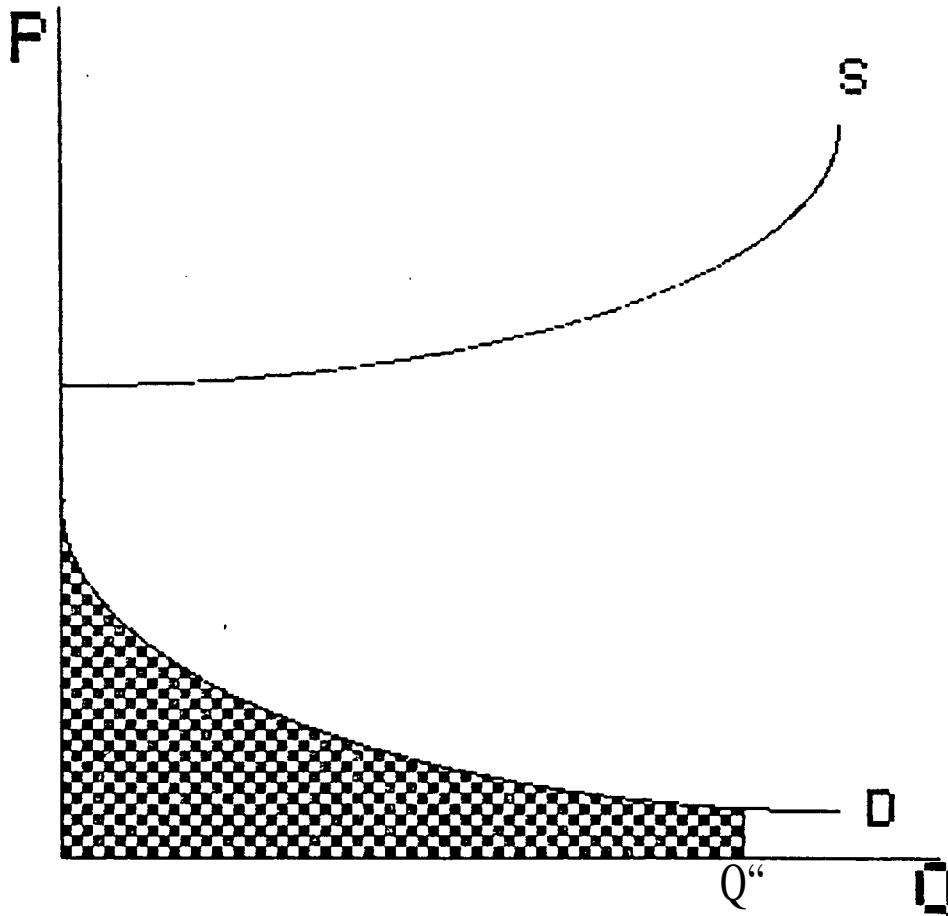## BENEFIT PROVIDED BY
THE ECOSYSTEM

FIGURE 3

WASTE ASSIMILATION
BENEFIT PROVIDED BY
THE ECOSYSTEM

This is essentially the difficulty with the pioneering and influential study of the value of estuarine wetlands by Gosselink, Odum, and Pope (GOP, 1974), They claim that an acre of estuarine wetland provides benefits which would cost $2,500 per year if produced by man-made treatment plants. Shabman and Batie (1977) are justifiably critical of this figure:

> ". · · the use of alternative estimates should be governed by three considerations: (1) the alternative considered should provide the same services; (2) the alternative selected for the cost comparison should be the least-cost alternative; and (3) there should be sub-stantial evidence that the service would be demanded by society if it were provided by the least-cost alternative. GOP failed to subject their estimate to any of these important tests."

Park and Batie (1979) contend that GOP not only failed to test whether the least-cost alternative would be demanded, but that their identification of waste treatment plants as the least expensive alternative may be incorrect. They argue that recent evidence suggests that adjustments in agricultural practices (e.g., restriction on the application of fertilizers which "run off" into estuarine waters) may be a less costly alternative to the construction of treatment plants. The criticism of the work of GOP is not to suggest that waste assimilation is not an important service provided by wetlands; however, care must be taken when determining just how society values that service.

Problems have also plagued efforts to value benefits which might be pro-vided by aquatic ecosystems sometime in the future but which are not currently provided. Instead of valuing the option to use a resource as an input to pro-duction in the future in the way suggested in Chapter 5, some studies have calculated benefits as if the resource were already being used. What is miss-ing here is an estimation of the likelihood that the resource will ever be

used and the timing of its use. Gupta and Foster (GF, 1976) attempt to value wet lands as a potential source of water supply for the state of Massachusetts and find that the state's wetlands could provide an annual benefit of $2,800 per acre. Unfortunately, GF's estimated benefit of wetlands' preservation in this regard is calculated as though the cost savings of using wetlands instead of current sources were already realized. Their finding, that wetlands would provide a cheaper supply of water for Massachusetts, can be questioned in two respects. First, if wetlands are a cheaper alternative to current sources, why are they not used? Second, if it is the existence of institutional barriers which block their use, why won't those barriers continue to preclude the tapping of wetlands as a supply of water in the future? Although it is certainly true that the preservation of wetlands may be valuable because the option to use them as a water source would be retained, this is not the benefit GF estimate. As a final point, their estimate of the total value of undeveloped wetlands may be plagued by double counting problems. If water were taken from Massachusetts' wetlands, would the same wetlands continue to generate the recreational and amenity benefits they add to the water supply benefits?

We now turn to the commercial harvest example. A substantial amount of previous empirical work has sought to value the environment as input for this purpose in ways not fully consistent with the deceptively simple approach discussed thus far and summarized in Figure 1. The estimated benefits variously fail to analyze changes in the relevant cost structure, ignore price effects of a change in production, and rely on ad hoc measures like total or net revenue. As a measure of change in social welfare, revenue figures exhibit at

least two problems. First, they do not reflect the opportunity cost of pro-
ducing goods and services. Second, demand for many fish and shellfish species
is relatively price inelastic (Bell, 1970), so an increase in production due
to an environmental improvement results in a decrease in total revenue, incor-
rectly implying that the improvement does not lead to a welfare gain. About
the best that can be said for the revenue calculations (with or without price
effects) is that they are not relevant to the determination of a change in
combined consumer and producer surplus--our preferred welfare measure.

A Council on Environmental Quality (CEQ, 1970) study illustrates the same
difficulties in a somewhat different way. The study reports that, due to the
practice of ocean dumping, one-fifth of the nation's shellfish beds are con-
taminated and closed. Assuming the closed shellfish beds would be as produc-
tive as their open counterparts, the study concludes that an improvement in
water quality would result in a 25 percent increase in quantity produced and a
subsequent 25 percent increase in total revenues. The increase in total reve-
nues are claimed as the gain to society of cleaning up the shellfish beds.
However, as long as demand is not perfectly elastic, an additional 25 percent
in the amount of shellfish supplied to the market could only be sold if the
price of shellfish fell. The estimate of CEQ of an additional $63 million in
shellfish revenues (the additional 2S percent) is clearly an overstatement.
But in any case the revenue figures do not reflect costs or willingness to pay
for nonmarginal units and, hence, are not adequate measures of welfare.

An important question to address, in valuing commercial fishing benefits,
is this: What is the contribution of the ecosystem to the production proc-
ess? It is a question some studies have failed to address. Thus, GOP (1974),

in assessing the value of wetlands as a fish nursery, divide annual dockside values of fish products landed by the total number of wetland acres to arrive at a value per acre in production of fish. Imputing all of the revenue from commercial fishing to wetland acreage, however, ignores the contribution of other fishing inputs like labor and capital.

The more recent study by Lynne, Conroy, and Prochaska (LCP, 1981) suggests that it may be possible to isolate the contribution of environmental inputs to production. They develop a bioeconomic model in which human effort and marshland are distinct inputs in the production of blue crab off Florida's Gulf Coast. The population of blue crabs is assumed to be a function of the quantity of local marshland acres. Since the successful harvesting of the crabs is modeled to be dependent on their population level, marshlands, which act to define the carrying capacity for blue crabs, appear as an input in the production function. The reduced form production function is estimated according to the ordinary least-squares criterion; and, using the appropriate estimated coefficients, a marginal product for an acre of wetlands is calculated. Finally, the value of the marginal product for an acre is computed using current dockside prices. The study is laudable for valuing both marshland acreage and human input in the production of blue crabs. However, the authors' contention that the value of the marginal product is the relevant measure of benefits provided by wetlands is incorrect. Let us take up the analysis at this point and develop an example in which notions of consumer and producer surplus are correctly employed, as in Figure 1, to evaluate the commercial fishing benefits produced by the marshland.

In keeping with the spirit of LCP, consider the optimization problem faced by a price-taking firmer industry where price is P and the unit cost of the human effort input is W:

(1) $\max_1$ P $F(X_1, \overline{X}_2) - W X_1$.

The production process is posited to be a function, $F(*)$, of two inputs: one (X_1) which captures the efforts of man to harvest shellfish and another (X_2) which represents the contribution of an ecosystem variable like marshland acreage. The bar over X_2 indicates that, for the time being, the acreage is fixed. Although we, like LCP, model human effort as a single input, the number of traps set, one many prefer to explicitly model the use of several inputs so that substitution among them can be studied.

We assume that the production of blue crabs can be represented as a Cobb-Douglas process. Although the Cobb-Douglas form is no doubt a simplification of the true production process (and is probably a poor approximation to reality for extreme values of either input), we use it here because our main purpose is to demonstrate the procedure for calculating changes in combined consumer and producer surplus. Therefore, substituting for the production function in equation (1) the Cobb-Douglas form and noting that cost minimization is the dual problem to profit maximization, the optimization problem can be rewritten as

(2) $\min_{X_1, \lambda} \mathscr{L} = W X_1 + \lambda(Q - A X_1^a \overline{X}_2^b)$

where λ is the Lagrange multiplier; Q is output; and A, a, and b are parame-

ters. Differentiating the Lagrangian with respect to the effort var.iable and

the Lagrange multiplier yields

(3) $\quad \dfrac{\partial \mathcal{L}}{\partial X_1} = W - \lambda A \, \bar{X}_2^b \, a \, X_1^{a-1} = 0$

(4) $\quad \dfrac{\partial \mathcal{L}}{\partial \lambda} = Q - A \, X_1^a \, \bar{X}_2^b = 0.$

Since the production function is characterized by only one decision variable,

$_1$, equation (4) is the only one needed to solve for the cost function,

c(*).

(5) $\quad X_1 = \left[\dfrac{Q}{A \, \bar{X}_2^b} \right]^{1/a}$

(6) $\quad C(W, Q, \bar{X}_2) \quad = W A^{-1/a} \, \bar{X}_2^{-b/a} \, Q^{1/a}.$

Differentiating the cost function with respect to output generates the mar-

ginal cost expression

(7) $MC = \dfrac{\partial C}{\partial Q} = \dfrac{W}{a} A^{-1/a} \, \bar{X}_2^{-b/a} \, Q^{(1-a)/a}$

The blue crab industry also presumably faces a demand curve for its product.

A simple constant elasticity demand function is given in (8), and the corres-

ponding inverse demand function in (9):

(8) $Q = KP^{-m}$

(9) $P = K^{1/m} Q^{-1/m}$

where K is a parameter and m is the (constant) elasticity. The **profit**-maximizing firms will equate price and marginal cost so that the equilibrium level of blue crabs sold is given by

(10) $Q = \left[\dfrac{a}{w} K^{1/m} A^{1/a} \bar{X}_2^{b/a} \right]^{ma/[m+(1-m)a]}$.

The result in (10) holds for all relevant values of marsh acreage, $\bar{X}_2$, available for the biological promotion of the blue crab population. There-fore, we first calculate the equilibrium output associated with various levels of wetland acreage, then we compute the equilibrium price corresponding to the output by use of equation (9).

We proceed to calibrate the parameters of the model in order to construct an example which is reasonably compatible with the price, input, and output data used by LCP. We also incorporate their econometric finding that the mar-ginal product of an acre of marsh is roughly 2-1/2 pounds of blue crab (annu-ally). Although the demand for shellfish has been found to be relatively price-inelastic, as we noted earlier, we assume in this case a high elasticity since the Gulf Coast fishery is presumably not the sole source of blue crab in the market. Welfare gains associated with an increase in marshland habitat (remember we are considering only gains in the blue crab industry for purposes of this example) are calculated as the change in consumer and producer sur-plus. These measures are presented in Table 1. For example, for a demand elasticity of -2.05, the net gain associated with an increase from 25,000

TABLE 1

Welfare Gain Associated with an Increase in Wetland Acreage
(From an Initial Base of 25,000 Acres)

Elasticity (m)	Wetland acreage $(\overline{X}_2)$	Number of traps (X_1)	Change in combined surplus
2.05	100.000	33.610	191,389
2.05	200,000	33,332	294,290
2.05	300,000	33,170	356,843
2.05	400,000	33,056	402,316
2.05	500,000	33,000	435,829

acres to 100,000 is $191,389. Successive increments in acreage add less to estimated benefits due to diminishing returns to the marshland input.

The results of a sensitivity analysis, in which different price elasticities of demand [ranging from (- .25) to (-3.45)] are used to calibrate the model, indicate that, in this particular model, the estimates of welfare gain are reasonably robust to the choice of an assumed price elasticity.

The purpose of this exercise has been to demonstrate that a theoretically correct measure of welfare can be constructed and calculated on the basis of empirical information about the impact on product supply (given demand) of a change in ecosystem characteristics (here the number of wetland acres) which, in turn, might be related to pollution control.

Of course, this has been a hypothetical exercise; and, in an actual case study, one would econometrically estimate the demand and production functions necessary to conduct the welfare analysis. Moreover, if the estimated demand function includes an income variable, simple Marshallian consumer surplus is no longer the appropriate welfare measure. Fortunately, for a variety of functional forms for the demand function, exact surplus measures are known and available.

A still more recent study, by Kahn and Kemp (KK, 198S), appears to follow the procedure we have outlined, though they use it to calculate a welfare loss. Specifically, they are concerned with the effect the decline in submerged aquatic vegetation (SAV) is having on the various fisheries supported by Chesapeake Bay. SAV serves as an important link in the estuarine food chain, and KK attempt to quantify the welfare loss primarily to the striped bass commercial fishery and, also, to other commercial and sport fisheries stemming from the reduction in SAV caused by agricultural runoff, discharges

from sewage treatment plants and soil erosion, and the consequent reduction in the carrying capacity of the Bay. Unlike LCP, KK are fortunate to have population data on the striped bass. With this, they can estimate a supply function which includes a population variable for the fish and an equation which relates SAV to fish. After estimating a demand function for striped bass, KK calculate the losses in consumer and producer surplus following incremental reductions in SAV. One criticism that can be made of their procedure is that, since demand is estimated as a function of per capita income, a more exact welfare measure than Marshallian consumer surplus could have been calculated. Just for purposes of comparison with the welfare gains that we calculated for the Florida Gulf Coast blue crab fishery, we observe that a 50 percent reduction in SAV is associated with an annual loss of approximately $4 million. This is substantially larger than the numbers in our example. It is important to note that KK are casting a wider net, so to speak: both commercial and sport fishing, for several species, are considered.

The studies just described are limited by their static nature. Both examine the contribution of an environmental input to production assuming the fishery is in bioeconomic equilibrium (i.e., the harvest rate of the marketed species equals its growth rate). To the extent that their data are comprised of observations for years in which the fisheries were not in a steady state, the regression coefficients they obtain will be biased as parameters of steady-state models. In addition, static approaches to fisheries economics fail to evaluate the stream of benefits generated by fisheries as they move from one equilibrium to the next. As demonstrated in Chapter 3, the higher

trophic levels of damaged ecosystems may respond slowly to pollution control measures, and attempts to value control need to take this into account.

The need for dynamic analysis arises from the recognition that fishery resources constitute capital assets which yield a stream of benefits over time, and it is in this framework that we can view proposed environmental cleanup policies as potential investments. Although much of the literature now recognizes the dynamic nature of fishery resources, with a few articles even explicitly recognizing the dynamic links between predator and prey species (see Clark, 1976, and Ragozin and Brown, 1985), the literature has not considered the management of fisheries' environmental problems in a dynamic context.

A framework for finding an optimal management strategy when a fishery is confronted with pollution and open-access problems might look something like the following. The management problem is one of simultaneously determining harvesting and pollution control policies to maximize the present discounted value of net benefits generated by the fishery. In the most general notation, i.e., making no assumptions about the forms of economic or biologic functions, the management problem is

$$(11) \quad \max_{E(t), Z(t)} \quad \sum_{t=0}^{\infty} (1 + r)^{-t} \, NB[E(t), Z(t), X(t)]$$

subject to

$$(12) \quad X(t+1) - x(t) = f\{E(t), Q[Z(t)], X(t)\}$$

and

$$(13) \quad x(o) = X_0$$

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where r is a discount rate, NB(·) is a net benefit function (e.g., combined consumer and producer surplus), E is fishing effort, Z is pollution control, X is the stock of the harvested species, and Q is the level of environmental quality. Further realism may be given to the model by including additional equations of motion [like equation (12)] which represent the growth rates of other species in the ecosystem and establish links between distinct levels of the food chain. Modeling species interaction may be of particular importance if pollution directly affects growth rates at the lower trophic levels, as demonstrated in Chapter 3. However, the introduction of biological interaction among species also poses the problem of selecting an appropriate model from the available alternatives (see May, 1973, for a description of the various ways in which species interaction may be modeled). Interactions can be complex and models like the Lotka-Volterra used in Chapters 3 and 4 and also in the studies reviewed in this section which imply simple feeding hierarchies rather than complex food webs may be misleading (see Harte, 1985).

A key feature of the solution of the optimization problem stated in equations (11) through (13) may be the interdependence of the two control variables, allowable fishing effort, and pollution control. For example, if the level of the fish stock is below the optimum, the derived solution to the management problem may include the enactment of stringent pollution controls to enable the fish population to recover. The solution may also include concurrent restrictions on fishing effort (possibly even prohibition) so that the eventual benefits of costly pollution control may be realized.

The fisheries management problem is further complicated by the fact that decisions must be made in the face of uncertainty. As discussed in Chapter 4,

uncertainty pervades the modeling of species interaction; and this is compounded by uncertainty about ecosystem responsiveness to pollution control. When uncertainty about the values of economic variables is introduced, the optimization problem becomes a very difficult stochastic control problem indeed. If it is the case that uncertainty about the parameters of the model can be reduced by research or the acquisition of information through experience, management strategies should ideally be evaluated with the aid of closed-loop models in which policy decisions are subject to revison as new information becomes available, as discussed in Chapter 5 (see also Rausser, 1978).

II. Aquatic Ecosystems as a Final Good

When an aquatic ecosystem is conceived ofas a final good the benefits of enhancing the ecosystem typically take the form of improved opportunities for water-related recreation. These benefits can be estimated using the methodologies discussed in Chapter 6--either contingent valuation/behavior experiments or the revealed preference approach based on fitting demand functions for visiting alternative recreation sites (also called the "travel-cost" approach). Some of the methodological issues involved in contingent valuation experiments are discussed in Cummings, Brookshire and Schulze (1986), Hanemann (1985), and Carson and Mitchell (forthcoming). Issues involved in the travel-cost approach are discussed in Bockstael, Hanemann and Strand (1984) and Smith and Desvousges (1986).

The main challenge confronting practitioners of travel-cost studies is the need to handle the allocation of water-based recreation activities among multiple sites differing in their environmental quality attributes in a manner consistent with the utility maximization hypothesis. Two particular aspects

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stand out--the selection of appropriate functional forms for the ordinary

demand functions, and the need to deal with corner solutions. Taking the

question of functional forms first, the problem is to select a set of functions

for the ordinary demands, xi = h^1 (p, q,y) , i = 1, . . . N, defined at the be-

ginning of Chapter 6. In this context xi is the number of visits to re-

creation site i by a household over some period of time (e. g. , the fishing

season) , p = (p_1,..,p_N) where pi is some measure of the cost of visiting

the ith site, q = (q_1,.., q_N) where q is some vector of attributes of the

i^{th} site (including water quality, etc.) and y is either the household's

total income or its total expenditure on recreation activities. The problem

is that, if these demand functions are to be consistent with some utility

maximization hypothesis, they must satisfy certain economic integrability

conditions, including (i) the adding up condition and (ii) the symmetry and

(iii) negative semidefiniteness of the matrix of Slutsky terms, $S = \{ s_{ij} \}$, where

$$S_{ij} = \frac{\partial h^i(p,q,y)}{\partial p_j} + h^j(p,q,y) \frac{\partial h^i(p,q,y)}{\partial y} . \tag{14}$$

These requirements are by no means trivial and impose significant

restrictions on the eligible functional forms. For example a demand system

of the form

$$\ln x_i = \alpha_i - \beta_i p_i + \gamma_i y \qquad i=1,..,N \tag{15a}$$

where
$$\alpha_i = \alpha_0 + \sum_k \delta_{ik} q_{ik} \tag{15b}$$

$$\beta_i = \beta_0 + \sum \delta_{2k} q_{ik} \tag{15c}$$

$$\gamma_i = \gamma_0 + \sum \delta_{3k} q_{ik} \tag{15d}$$

which is employed in Smith and Desvousges (1986) , would appear to violate

the symmetry of the s_{ij} terms. Other generalizations of the semi-log form

to systems of multiple demand equations are examined by Hanemann and Lafrance

(1983) , where it is shown that the symmetry conditions place very stringent

(and empirically implausible) restrictions on the underlying direct utility
function.   This does not mean that there are no suitable functional forms:
systems such as the Linear Expenditure System-- Binkley and Hanemann (1975) --
and other members of the Generalized German Polar Form family of indirect
utility functions

$$v(p,q,y) = F\left(\frac{y}{b(p,q)}\right) + a(p,q) \tag{16}$$

can certainly be employed.

The second issue--the phenomenon of <u>corner</u> solutions--is more trouble-
some.   This refers to a situation where some of the xi's are zero--a household
visits some of the available sites, but not all of them.   The conventional theory
of consumer behavior is developed under the assumption of an <u>interior</u> solution
to the utility maximization problem   (1) in Chapter 6--i. e. , a solution where
all the xi's are positive.   Modifying this theory to deal with non-consumption
of certain goods (non-visitation of certain sites) --a phenomenon that is
overwhelmingly apparent in micro-data sets--is a rather complex task.   The
problems involved, and some possible solutions, are examined in Chapters
8-10 of Bockstael, Hanemann and Strand.

A common approach to modelling corner solutions is to decompose consumer
choices into two elements: the selection of a <u>total</u> level of recreation activity,
$\bar{x} = \sum x_i$, and then the allocation of this total among the alternative possible
sites based on some type of shares model

$$x_i = \pi_i (p, q, y) \, \bar{x} \qquad\qquad i = 1, \dots, N \tag{17}$$

where π_i, the share of total visits assigned to the ith site, satisfies

$$\pi_i \geq 0, \qquad \sum \pi_i = 1. \tag{18}$$

Statistical models such as logit and probit can be used to estimate the share
equations, and these models can be related to a utility maximization hypothesis.
But, at the present time, it is often difficult to obtain a utility-theoretic

justification for the "macro visitation equation" determining $\bar{x}$, and to integrate it with the share equations in a theoretically consistent manner. That is to say, one would like the determination of $\bar{x}$ and $\bar{n}_1$, . . . $\bar{n}_N$ to originate in a single, simultaneous utility maximization procedure. Some models which permit this have recently been developed, but they are relatively difficult to estimate. The resolution of these issues represents one of the frontiers of research for the travel cost approach.

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Chapter 8.

Further Work

Our present intention is to proceed in two areas: (1) comparative analysis of models for policy evaluation; and (2) development of a case study .

The **first task,** the comparative analysis, is intended to further integrate the ecologic and economic **models** developed in earlier chapters, and to compare the results obtained with those **of** variant versions of the models. Both aspects of this task are important. The first involves a tighter linking (than **any** in the Present report) of a model of ecosystem recovery with a model of dynamic optimization under uncertainty. The idea is to develop the capability to evaluate *control* policies leading to ecosystem recovery, taking account of the (probabilistic) state of the system over time and at any point in time.

The second aspect of this task, comparative analysis of different models, is dictated by our lack of knowledge about population dynamics in a recovering aquatic ecosystem. In chapters 3 and 4 these dynamics *were* described by perhaps the simplest model for the purpose, the Lotka-Volterra. This was sufficient to **obtain** interesting results about qualitative features of recovery dynamics and the propagation of uncertainty. But as we move toward application (as in the case study described below) it becomes important to determine whether the results are **robust,** i.e., whether they continue to hold for equally plausible, though more complex, specifications of ecosystem population dynamics. Further, we need to explore the notion of robustness itself. Two models may yield seemingly quite different predictions about the nature and timing of recovery, yet imply the same

ranking of policy alternatives. For example, one model may predict recovery of a fish Population to 50% of itspre-pollutionlevel (ignoring uncertainty) within five years of the imposition of some control measure, whereas another may predict recovery to just 10%. But the net present value of control may be positive in both cases. In any event, considerable further work is needed, in our judgment, on model development, integration, and comparative analysis, before we are ready to tackle a case study.

Turning now to the case study, we wish to pose a basic question: What do we want to get out of a case study? Two things, it seems to us. First, of course, we want quantitative results. What are the benefits of a particular control option? Second, however, we want to know what the results depend on. Partly, this is traditional sensitivity analysis. How are results affected by changes in assumptions about the discount rate, about a parameter describing interaction between the first and second trophic levels, and so on. But more importantly, we want to try to establish links between results and the types of models used to generate them. This task clearly links back to our proposed work in the first area, comparative analysis of models for policy evaluation. The difference is that now we are proposing to go through the exercise in a real case, with real numbers.

With these objectives in mind, we wish to propose a 'double-barreled" study. First, we would look at a relatively simple lake ecosystem, and one for which there also exists fairly good data on pollution control and subsequent recovery. A leading candidate here is Lake Washington, in the state of Washington. The idea would be to "field-test" our modeling approach in a relatively favorable setting.

Second, we would like to tackle San Francisco Bay. The Bay is of

course a much **larger** and more complex **aquatic** ecosystem, a marine estuary with substantial wetlands. Further, existing data are less reliable than for Lake Washington. Yet even with these difficulties, we feel the Bay is an appropriate subject for study by this **project,** for several reasons. First, it is economically important, a major influence on the natural resource base (including climate) of a metropolitan area of more than five million people. Second, the Bay **is** the subject of considerable current research and policy interest, at both the state and national levels. Third, a related point, the Bay ecosystem includes the major remaining wetlands in Northern California, and wetlands are themselves the subject of much current interest. Fourth, a study of San Francisco Bay would nicely complement existing work on the major east **coast** marine **estuarine system,** the Chesapeake Bay. Fifth, clearly travel costs would be minimized by choice of the Bay. Sixth, and finally, despite, or perhaps because of, the difficulties, we regard the proposed study as an exciting challenge.

We should note that, again because of the magnitude of the task and the potential difficulties, we do not propose to complete a study of the **Bay** within 12 to 18 months following submission of the final report on the current study. But we certainly would anticipate completion of parts of the task, which might stand on their own **as** interesting and useful research results .